Waste Management 81 (2018) 202–210

Contents lists available at ScienceDirect

Waste Management

journal homepage: www.elsevier.com/locate/wasman

Effects of different agricultural organic wastes on soil GHG emissions: During a 4-year field measurement in the North China Plain

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ARTICLE INFO

Article history:
Received 12 May 2017
Revised 4 September 2018
Accepted 5 October 2018
Available online 10 October 2018

Keywords:
Soil GHG emission
Organic wastes
Emission factors
Characteristics of organic wastes
Cropping system
North China Plain

ABSTRACT

Large quantities and many varieties of agricultural organic wastes are produced in China annually. Applying agricultural organic wastes to soil plays an essential role in coping with the environmental pollution from agricultural wastes, solving the energy crisis and responding global climate change. But there is little information available on the effects of different agricultural organic wastes on soil greenhouse gas (GHG) emissions. The objectives of this study were to investigate and compare the impacts of different organic wastes on soil GHG emissions during a 4-year field experiments in the North China Plain, as well as analyze the influential factors that may be related to GHG emissions. The treatments were: crop straw (CS), biogas residue (BR), mushroom residue (MR), wine residue (WR) and pig manure (PM) returning to soil, as well as a control with no organic waste applied to soil but chemical fertilizer addition only (CF). The results showed that compared with CF treatment, organic material applied to soil significantly increased GHG emissions and emissions followed the order of WR(27,961.51 kg CO_2-eq/ha/yr) > PM (26,376.50 kg CO_2-eq/ha/yr) > MR(23,366.60 kg CO_2-eq/ha/yr) > CS(22,434.44 kg CO_2-eq/ha/yr) > BR (22,029.04 kg CO_2-eq/ha/yr) > CF(17,402.77 kg CO_2-eq/ha/yr), averagely. And considering the affecting factors, GHG emissions were significantly related to soil temperature and soil water content. Different organic wastes also affected soil total organic carbon (TOC), microbial carbon (MBC) and dissolved organic carbon (DOC) contents, which related to GHG emissions. Further analysis showed that characteristics of organic wastes affected GHG emissions, which included C-N ratio, lignin, polyphenol, cellulose and hemicellulose. Our study demonstrates that biogas residue returning to soil emitted minimum GHG emissions among these different types of organic wastes, which provided a better solution for applying organic wastes to mitigate soil GHG emissions.

1. Introduction

Global climate change is arguably the most urgent environmental challenge confronting society. Atmospheric carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O) are the most potent long-lived greenhouse gases (GHG) that contribute to global warming. Agriculture, which contributes to about 10%–12% of total global anthropogenic GHG emissions (IPCC, 2007), is recognized as the important source of GHG emissions. CO_2 and CH_4 emissions in cropland are derived from a variety of practices in the agricultural sector, including soil tillage, soil drainage, rice management, biomass burning, flooding, and the use of fertilizers and residues (Smith et al., 2008; West and Marland, 2002). N_2O has a global warming potential 298 times greater than CO_2, over a hundred year time horizon, which is derived from soil microorganisms activities, such as nitrification and denitrification processes (IPCC, 2007), which can be reduced by the field management practice like, no-tillage (Van Kessel et al., 2012), straw return (Yao et al., 2017).

The livestock sector contributes 40–50% of agricultural GDP, but also should be responsible for greenhouse gas emissions of 5.6–7.5 Gt CO_2-eq/yr; and in order to mitigate global warming potential, it's important to reduce emissions from manures or wastes and

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https://doi.org/10.1016/j.wasman.2018.10.008
0956-053X/

to promote carbon sequestration (Herrero et al., 2017). Organic wastes have already been widely used in China, since they carry a full range of nutrients and are rich in biologically active substances (Zhao et al., 2009), which is not only helpful to improve soil fertility and crop quality but also beneficial to protect the environment. Adding them to the farmland, one the one hand, they can supply various macro-nutrients and micro-nutrients, improve soil chemical and physical properties and promote plant growth (Zhao et al., 2009); on the other hand, the microbial biomass carbon and enzymatic activity can be increased. And long-lasting application of organic wastes showed that they can increase the soil organic carbon (Diacono and Montemurro, 2010). Thus, organic residues can be used as soil organic amendments, which are available from a wide range of production process, such as crop production, animal production, food and energy processing and municipal sources.

Plenty of benefits applying organic wastes to soil have been reported as mentioned above, however, the emissions of GHGs become remarkable during the organic wastes production and application, such as the compost (Sommer et al., 2009; Leytem et al., 2011) and the incorporation of organic wastes to soil. Previous research mainly concentrates on the comparison of GHG emissions between certain kinds of organic wastes or between chemical fertilizer and organic fertilizers. Generally, organic wastes can certainly enhance GHG emissions compared with inorganic fertilizers (Ding et al., 2007; Verhoeven and Six, 2014). Nonetheless, via a more comprehensive and useful method-life cycle assessment (LCA), which can analyze potential environmental impacts throughout a product's life cycle including the supply chain and downstream processes (ISO, 2006), some researches point out that GHG emissions avoid the consequence of the "no" use of chemical fertilization. For example, Moore et al., (2017)reported that replacing chemical fertilizers with vinasse and filter cake is beneficial to the environment since it reduces GHGs emissions in the entire process of ethanol production.

In addition, a large amount of agricultural wastes is produced in China not only with tremendous quantity but also with multiple species. According to National Development and Reform Commission of China (NDRC), the theoretical volume of crop straw resources reached to 863 million tons in 2011, and the annual livestock manure production was estimated to be 3 trillion tons, but quite a number of them were wasted or non-recycled (NDRC, 2012). Besides, except the common wastes such as crop straw and livestock manure, with the development of diversified agricultural production, diverse organic wastes were produced in China, such as biogas residues, mushroom residues and wine residues. Nowadays, biogas has become an important component of sustainable development because it's attributable to the reduction of environment pollution and the alleviation of energy shortage effectively. By the end of 2015, the total biogas yield was 15.8 billion m^3 in China which means that a great deal of biogas residue has been produced equivalently (NDRC, 2017). Food and Agriculture Organization (FAO) estimated the production of mushrooms and truffles reached to 6.4 billion t in 2008 and China contributed more than 70% of world's total production of mushrooms, which also created huge mushroom residues (FAOSTAT, 2012). And due to the longtime drinking history in China as well as the growth of population, it brings not only the increase of wine consumption, but also the tremendous wine residue from brewing at the same time.

The area of North China Plain (NCP) is about 14 million hectare located in Hebei, Shandong, and Henan Provinces (account for 1.5% of the country), in which the area of approximate 7.7 million hectare are currently under cultivation (account for 6.4% of the country); and the population is approximate 120 million (account for 8.7% of the country). As an important center of agricultural production, it accounts for 61% of the country's wheat production and over 30% of the maize production in China. Meanwhile, the massive grains production of brings a lot of agricultural wastes. Since we know that there are indeed many benefits that organic wastes as soil amendments can bring to us, Chinese government encourages farmers to reuse and recycle wastes in the agricultural production to achieve the environmental-friendly and sustainable development of agriculture. However, there is still little scientific information about the effects of GHG emissions from different agricultural wastes. Thus, this paper aims at the investigation of GHG emissions and the global warming potential induced by different types of organic wastes after adding to the farmland. And in this study, through a 4-year experimental measurement with summer maize-winter wheat rotation, which is a mainly cropping pattern in NCP, the dynamic and total emission of main GHG, i.e. CO_2, N_2O and CH_4 were analyzed, along with their influential factors, which could also be functional in the construction of essential data base guiding for GHG mitigation in agriculture processes, especially for agricultural waste management.

2. Materials and methods

2.1. Experimental site

The experiment was conducted at the Wuqiao Experimental Station in Wuqiao County, Hebei Province (37°41′02″N, 116°37′23″E). The mean annual temperature at the site is 10–12 °C, sunshine duration is 2724 h and accumulated temperature (≥0 °C) is 4826 °C. The average annual rainfall during the last 20 years was 560 mm. Generally, 60–70% of the annual precipitation occurs from June to August, which covers most of the maize season. The soil at the experimental site is classified as alluvial salt aquic soil and classified as silty loam (IUSS Working Group WRB 2006). The initial texture and principal properties of experiment soil for 0–10 cm and 10–20 cm depth are available in Table 1.

2.2. Experimental design

The experiment was established in October 2012. Experimental plots (2 × 2 m) were arranged in a randomized complete block design with three replications per treatment. This micro-plot trial is beneficial to detect the impact factors of different organic wastes returning to field as precisely as possible. Winter wheat (*Triticum aestivum L.*, from the middle of October to early June) variety Jimai 22 was sown at a rate of 300 kg/ha, and summer maize (*Zea mays L.*, from early June to the middle of October) variety Zhengdan 958 was seeded at a rate of 45 kg/ha. Following the fertilizer strategy of many longtime experiment of organic wastes, all plots received a broadcast fertilizer application of P_2O_5 26 kg/ha (diammonium phosphate) and K_2O 124 kg/ha (potassium sulfate) before seeding and pure nitrogen 150 kg/ha (urea) before seeding and at the jointing stage separately in every growing season.

There were five treatments of different organic wastes returning to fields: crop straw (CS), biogas residue (BR), mushroom residue (MR), wine residue (WR) and pig manure (PM), and one treatment with only chemical fertilizers (CF). Although in actual farming practices, nutrition supply, such as nitrogen, also is an important topic, since we wanted to focus more on soil carbon storage, soil GHG emissions and the entire carbon cycle of croplands under the circumstances of organic wastes inputs, the amount of organic wastes returned to the field should be equivalent as far as carbon amount in every growing season, which means that organic wastes carbon was needed to be returned as the same amount as the harvesting, based on themselves carbon content and quantities. Therefore, we calculated that the actual amount of C

Table 1
Soil texture and basic properties.

Soil depth (cm)	Soil texture			Soil principal properties			
	Sand (>0.05 mm)	Silt (0.05–0.002 mm)	Clay (<0.002 mm)	Organic carbon (g kg^{-1})	Bulk density (g cm^{-2})	EC (ms cm^{-1})	pH
0–10	24	60	16	4.5	1.41	0.91	8.2
10–20	31	56	13	4	1.47	0.66	8.2

inputted in the winter wheat season of 2012–2016 was 3054 kg C/ha, 3768 kg C/ha, 3694 kg C/ha, 2603 kg C/ha and 2603 kg C/ha, respectively; and in the summer maize season was 2683 kg C/ha, 2930 kg C/ha, 4040 kg C/ha, 2296 kg C/ha and 3585 kg C/ha, respectively.

2.3. Determination of soil GHG emissions

Soil GHGs (CO_2, CH_4 and N_2O) were collected by static cylindrical chambers (25-cm height and 25-cm diameter), which is a widely used method adopted by many researchers (Ju et al., 2011; Huang et al., 2014). The chamber was covered with 5-mm-thick acrylic material in order to reduce the impact of directly thermal radiation during gas sampling. And there were two battery-operated fans inside the chamber to mix the gas. The collar-like foundations of chambers were placed between the crop rows, inserted 5 cm into the soil, and maintained inside clearance without crops and weeds all the time. After putting chambers onto the foundations and sealing them using water, gas samples were collected by a plastic syringes (50 mL) attached to a three-way stopcock at 8, 16 and 24 min intervals and then transferred into a pre-evacuated glass exetainer (12 mL). The frequency of gas samples' collection was once per week ordinarily and added more time in the following conditions: rainfall and application of nitrogen fertilizer. More specifically, we collected gas samples after rainfall or irrigation as soon as possible and after 1, 3, 5, 8 and 13 days of application of nitrogen fertilizer in order to grasp the peak of the soil GHG emissions. All of the measurements were conducted in the morning (09:00–10:00) (Tian et al., 2012).

Gas samples were analyzed immediately by a gas chromatograph (Shimadzu GC-2014C). And then we calculated the gas flux rates F (mg/m^2/min) with Eq. (1) and the cumulative amounts E with Eq. (2) (Javed et al., 2008).

$$F = \frac{M}{Mv} \times \frac{V}{A} \times \frac{dc}{dt} \times \frac{To}{T} \tag{1}$$

$$E = F \times D \times \alpha \tag{2}$$

where M refers to the molar mass of N_2O (44 g/mol), CO_2 (44 g/mol) and CH_4 (16 g/mol); Mv (L/mol) is the molar volume of them; V (m^3) and A(m^2) are the volume and bottom area of the chamber, respectively; dc/dt means the linear changing rate of GHG concentration; To (273.2 K) is the absolute temperature; T (K) is the air temperature inside the chamber; D (d) means the number of days between sampling dates; α refers to the global warming potential (GWP) factors of N_2O (298 CO_2-eq), CO_2 (1 CO_2-eq) and CH_4 (25 CO_2-eq) (IPCC, 2007).

2.4. Determination of factors related to GHG emissions

Soil carbon components, soil temperature and soil moisture are important exogenous factors affecting GHG emissions, while the properties of organic wastes also are the essential endogenous factors which can be applied to explain the difference of GHG emissions between treatments.

Soil temperature moisture were measured with gas sampling simultaneously. Soil temperature was measured to a depth of 5 cm using soil thermometers approximately 10 cm apart from the chambers. Soil moisture was measured by Time Domain Reflectometry (TDR) and the depth of soil moisture content was 0–5 cm.

Soil samples were collected after harvesting using five-spot-sampling method in each plot with a soil augur (2 cm diameter) in the depth of 0–10 cm and 10–20 cm, then mixed and air dried after removing the visible roots and crop residues. The total organic carbon (TOC) was determined using an exogenous thermal process with potassium dichromate (Walkley and Black, 1934). The microbial carbon (MBC) was determined using substrate induced respiration method (Lin and Brookes, 1999). The dissolved organic carbon (DOC) were adopted the method of Bolan et al. (1996).

As for the properties of organic materials, carbon content was measured using an exogenous thermal process with potassium dichromate (Walkley and Black, 1934); Nitrogen content was estimated using the Kjeldahl method; and the concentration of cellulose, hemicellulose, lignin and soluble substance were analyzed using the Van Soest acid detergent fiber method (Van Soest, 1963); ash was measured by combustion method and polyphenol was determined using the ferrous tartrate method (Turkmen et al., 2006).

2.5. Statistical analyses

The data was initially collated with Excel 2010 (Microsoft, Redmond, WA, USA). Statistical analyses were performed using SPSS 17.0 (SPSS, Inc., Chicago, IL, USA). Significant differences in cumulative soil GHG emissions, average soil temperature, soil moisture, TOC, MBC and DOC between treatments were identified using analysis of variance by least significant difference calculations. GHG flux data were log-transformed. Multiple comparisons were made with Duncan's multiple range tests, significant differences between treatments were determined at 95% level probability. To reduce uncertainty, for GHG emissions, spectrometer allowed an extremely sensitive detection of a wide variety of atmospheric trace gases with high precision and accuracy. High variability in soil emissions due to geographic location and site-specific weather conditions. Thus, the probability of type I errors was countered against the increasing probability of type II errors (i.e., accepting the null hypothesis when false, or failing to declare a real difference as significant). We report least squared means and standard errors. Correlation analysis was conducted and a $p < 0.05$ was considered significant.

3. Results and discussions

3.1. Soil GHG emissions

3.1.1. Effects of different organic wastes on soil CO_2 emissions

Like other results of research, as the most integral part of greenhouse gases, CO_2 emission accounted more than 90% of the total emissions and our results shown that it took responsibility for 96.55–98.50% of GHGs emissions in the treatments of organic materials. The readily-decomposable organic carbon compounds in organic wastes may trigger greenhouse gas emissions by enhancing respiration and providing energy for denitrification

(Cayuela et al., 2010). Compared with chemical fertilizers (CF) treatment, all of the organic wastes significantly promoted the CO_2 emissions ($P < 0.05$). Four-year average results showed that CO_2 emissions from crop straw (CS), biogas residue (BR), mushroom residue (MR), wine residue (WR) and pig manure (PM) treatments were higher 29.37%, 26.29%, 34.11%, 58.05% and 49.27% than that of CF treatment, respectively (Table 2). CO_2 emission in the soil is the sum of the autotrophic respiration, derived from root system of plants and the heterotrophic respiration, derived from soil organisms, such as microbes and soil fauna (Hanson et al., 2000). Thus, the difference of soil CO_2 emission between organic wastes was due to the complex interactions existing between soil microbes, soil fauna and plant root system after organic wastes was applied to soil.

Additionally, there were obviously seasonal variations of CO_2 flux. Four-year results consistently showed the fluxes values of maize season were much higher than that of wheat season (Fig. 1a). Organic material incorporation triggers the activity of soil microorganisms and increased soil aggregation, which causes the acceleration of microbial metabolism and the increase of their respiratory activity (Thangarajan et al., 2013). Therefore, higher environmental temperature induced higher microbial activity and then caused greater CO_2 emission. The four-year average cumulative CO_2 emissions of organic waste treatments were higher 25.58% than CF treatments in wheat season and 47.96% in maize season, respectively. And the peak of CO_2 fluxes was captured after irrigation and fertilization. The values of CO_2 flux were between −1.21 and 34.19 mg/m^2/min.

Table 2
Global warming potential (GWP) under different organic wastes incorporation (kg CO_2-eq/ha).

Years	Seasons	GHG	CS	BR	MR	WR	PM	CF
2013	Wheat	CO_2	9773.77a	7649.29ab	8089.15ab	9784.89a	9770.51a	7203.84b
		CH_4	−5.48a	−17.79a	−20.14a	−9.48a	−5.07a	−11.65a
		N_2O	137.57b	140.6b	75.57b	359.83a	311.63a	125.38b
		GWP	9905.86ab	7772.09bc	8144.58abc	10135.24a	10077.07a	7317.57c
	Maize	CO_2	13993.98b	14109b	15612.85b	16933.14ab	19265.36a	10751.97c
		CH_4	−6.18a	−6.47a	−6.53a	−5.84a	−8.76a	−8.28a
		N_2O	271.05c	513.53bc	753.43ab	1058.88a	732.33abc	442.35bc
		GWP	14258.84c	14616.05c	16359.75bc	17986.18ab	19988.94a	11186.04d
	Sum	CO_2	23767.75bc	21758.28c	23702bc	26718.03ab	29035.87a	17955.81d
		CH_4	−11.66a	−24.27a	−26.67a	−15.32a	−13.82a	−19.93a
		N_2O	408.61d	654.13bcd	829bc	1418.71a	1043.96ab	567.72cd
		GWP	24164.7b	22388.14b	24504.32b	28121.42a	30066.01a	18503.61c
2014	Wheat	CO_2	9676.03b	8026.3c	9375.44b	11246.54a	9911.24b	6265.65d
		CH_4	−14.85ab	−15.11ab	−11.72ab	−23.06b	−2.94a	−6.67a
		N_2O	332.78b	307.93b	352.38b	443.07a	357.91b	197.52c
		GWP	9993.96b	8319.13c	9716.1b	11666.55a	10266.21b	6456.5d
	Maize	CO_2	13254.67b	12031.58bc	14006.76b	17462.59a	14027.06b	9244.93c
		CH_4	−4.92a	−10.14a	−9.47a	−7.89a	−9.9a	−4.88a
		N_2O	196.76c	227.14bc	327.22b	491.31a	341.3b	167.31c
		GWP	13446.51b	12248.57bc	14324.51b	17946.02a	14358.46b	9407.37c
	Sum	CO_2	22930.7bc	20057.88c	23382.2bc	28709.13a	23938.31b	15510.58d
		CH_4	−19.77a	−25.25a	−21.18a	−30.95a	−12.84a	−11.54a
		N_2O	529.54c	535.07c	679.6b	934.38a	699.2b	364.83d
		GWP	23440.47bc	20567.7c	24040.62b	29612.57a	24624.67b	15863.87d
2015	Wheat	CO_2	8731.59a	6566.56b	7295.66ab	7595.14ab	6228.97b	6142.59b
		CH_4	−7.46a	−17.13a	−32.63a	−2.74a	−9.26a	−5.37a
		N_2O	166.06a	169.34a	140.3a	154.1a	339.68a	169.62a
		GWP	8890.19a	6718.77ab	7403.34ab	7746.5ab	6559.39ab	6306.84b
	Maize	CO_2	10320.21ab	10892.48ab	12251.17ab	13754.96a	10064.14ab	8583.16b
		CH_4	−5.23a	−10.88a	−9.59a	−16.74a	−1.32a	−8.48a
		N_2O	61.72bc	326.75ab	−4.27c	518.07a	318.53ab	92.2bc
		GWP	10376.69b	11208.35ab	12237.31ab	14256.28a	10381.35b	8666.88b
	Sum	CO_2	19051.8ab	17459.03ab	19546.83ab	21350.09a	16293.11b	14725.75b
		CH_4	−12.7a	−28.01a	−42.21a	−19.48a	−10.58a	−13.86a
		N_2O	227.77b	496.09ab	136.04b	672.16a	658.21a	261.82b
		GWP	19266.88ab	17927.12ab	19640.65ab	22002.78a	16940.74b	14973.72b
2016	Wheat	CO_2	5847.07a	6287.85a	6987.69a	8230.85a	6688.33a	6468.17a
		CH_4	0.09a	−9.55a	−19.77a	−16.11a	−19.35a	−18.18a
		N_2O	155.96a	110.52a	84.46a	188.15a	292.5a	102.78a
		GWP	6003.12a	6388.82a	7052.38a	8402.89a	6961.48a	6552.77a
	Maize	CO_2	16790.99c	20724.95 bc	18007.32bc	22978.5ab	26031.22a	13663.51c
		CH_4	−5.18de	−4.35bc	−5.26cd	−2.57ab	−2.79a	−9.41e
		N_2O	76.79ab	123.78ab	226.39ab	730.45a	884.64a	63.01b
		GWP	16862.6de	20844.37bc	18228.44cd	23706.38ab	26913.08a	13717.11e
	Sum	CO_2	22638.06cd	27012.79b	24995.01bc	31209.35a	32719.55a	20131.68d
		CH_4	−5.09a	−13.9a	−25.03a	−18.68a	−22.13a	−27.59a
		N_2O	232.75b	234.3b	310.85b	918.6a	1177.14a	165.79b
		GWP	22865.72bc	27233.19b	25280.82b	32109.27a	33874.56a	20269.89c
4 year average		CO_2	22097.08c	21572c	22906.51bc	26996.65a	25496.71ab	17080.95d
		CH_4	−12.3a	−22.86a	−28.78a	−21.11a	−14.84a	−18.23a
		N_2O	349.67b	479.9b	488.87b	985.96a	894.63a	340.04b
		GWP	22434.44b	22029.04b	23366.6b	27961.51a	26376.5a	17402.77c

Note: different lowercase letters in the same line of the same season indicate significant differences between different organic waste treatments ($p < 0.05$).

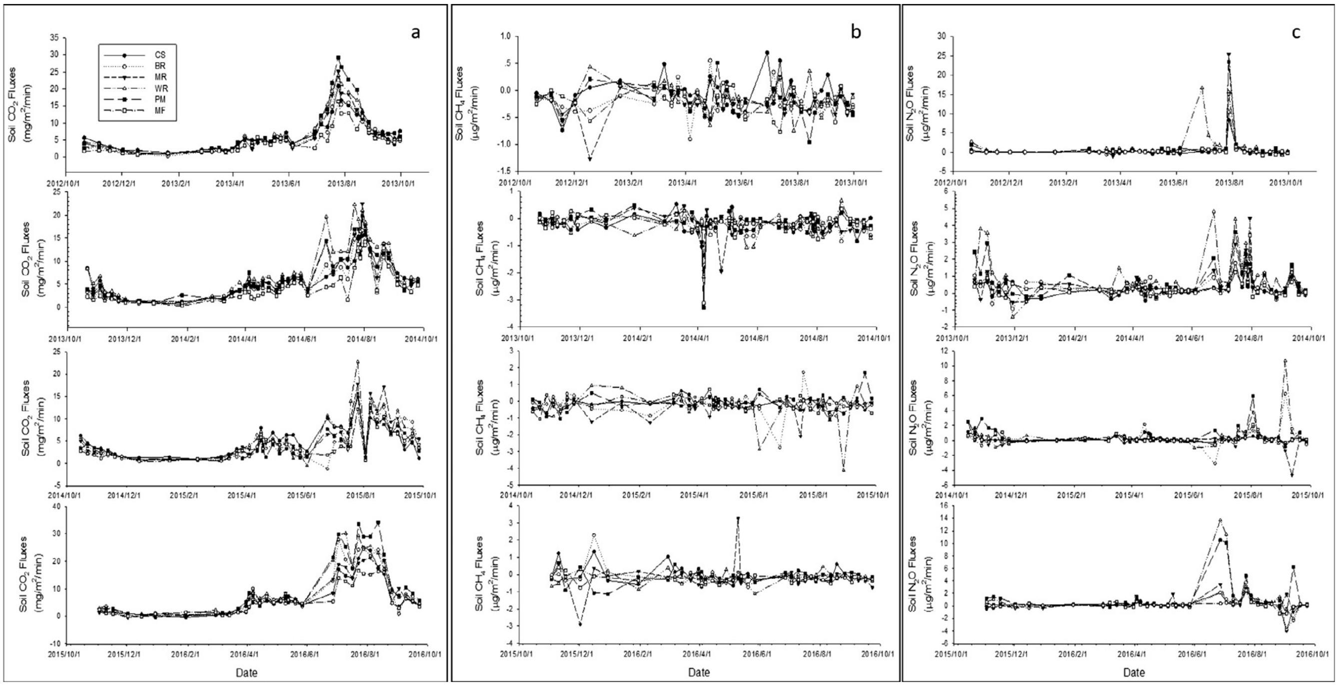

Fig. 1. Soil CO_2 (a) CH_4 (b) and N_2O (c) fluxes from 2012 to 2016.

3.1.2. Effects of different organic wastes on soil N_2O emissions

Organic wastes addition also promoted soil N_2O emission. Both nitrification and denitrification are sources of N_2O and they can occur simultaneously. The former process is dominated by aerobic conditions while the latter one is mainly caused by anaerobic microorganisms. Due to the input of organic wastes, increased the amount of readily decomposable organic components enhances the potential of nitrification and denitrification process. And as Nayak et al. (2015) reported that combined application of organic manure with N fertilizer can increase N_2O emission by 75% compared to NPK alone. Our four-year average results showed that N_2O emission from organic wastes treatments were 1.55 times and 2.14 times of CF treatment in wheat season and maize season, respectively. Results revealed that N_2O emission from WR and PM were significantly higher than the other treatments ($P < 0.05$) and the value of WR was the highest, which was 1.82, 1.05, 1.02, 0.10 and 1.90 times higher than CS, BR, MR, PM and CF (Table 2). This result is also in agreement with the observations of Rochette et al. (2008), who found that mineral fertilizer leads to less N_2O emission than cattle manure. Since significant denitrification can take place when NO_3–N and readily decomposable organic compounds are available (Bedard et al., 2006), N_2O emissions produced by denitrification would be more responsible for the soil GHG emissions.

The peak of N_2O fluxes appeared after fertilization, indicating that fertilization promoted soil N_2O emissions, especially in the maize season and ranged from −4.66 to 25.42 μg/m^2/min. And N_2O fluxes in maize season were higher than that of wheat season as same as CO_2 fluxes (Fig. 1c).

3.1.3. Effects of different organic wastes on soil CH_4 emissions

Soil CH_4 emissions in all treatments were functional as a tiny pool and MR treatment had the maximum CH_4 fixation value, but there were no significant differences between each treatment ($P < 0.05$). Waterlogged anaerobic rice paddies are a key source of CH_4 emission, which is the fourth largest agricultural source of GHGs (Smith et al., 2008). So most of the studies focused on CH_4 emissions from paddy field, CH_4 emissions from dryland are ignored because of it is insignificant for the total emissions (Nayak et al., 2015). From the results of 4-year winter wheat-summer maize cultivation, soil is a little bit of CH_4 sinks and CH_4 fluxes fluctuated around zero, from −4.11 to 3.26 μg/m^2/min (Fig. 1b).

3.1.4. GWP of different organic wastes

Based on the above analysis, applying organic wastes significantly affected soil GHG emissions, and the four-year average results showed that GHG emissions of different organic wastes treatments were significantly higher than that of CF treatment ($P < 0.05$). Among these five different organic wastes, GWP of WR and PM were significantly higher than the other 3 kinds of organic wastes ($P < 0.05$). The total GHG emissions were as follows: WR (27,961.51 kg CO_2-eq/ha/yr) > PM (26,376.50 kg CO_2-eq/ha/yr) > MR (23,366.60 kg CO_2-eq/ha/yr) > CS (22,434.44 kg CO_2-eq/ha/yr) > BR (22,029.04 kg CO_2-eq/ha/yr) > CF (17,402.77 kg CO_2-eq/ha/yr). Although the period of growing stage of winter wheat was much longer than summer maize, the accumulation of its GHG emissions was lower than summer maize (Table 2), which accounted for 61.23–67.90% GWP of the total year. CO_2 emissions accounted for the majority of GWP proportion among three kinds of gases. Although organic wastes can function as substitution of chemical fertilizers, working on improving soil conditions and supporting sustainable crop productivity, a large quantity of GHG emissions are reported by many studies and thus lots of researchers are working to seek solutions. Haque et al. (2016) reported that intermittent drainage and incorporation of 3 Mg/ha of green biomass could be a good management option to reduce GWP. And our results showed that biogas residue adding to farmland emitted minimum GHG emissions among these different types of organic wastes, which might be a better solution for selecting organic wastes to mitigate soil GHG emissions (Table 2).

Furthermore, it is worth noting that considering the soil GHG emissions merely cannot represent the overall GHG emissions or sequestrations. What's important is to analyze the net greenhouse gas balance with a more systematic and comprehensive perspective (Huang et al., 2013). As a successful holistic approach, life cycle assessment (LCA) has been wildly used in many research studies to

evaluate environmental impacts in agricultural production systems. However, most of the LCA tools were developed for general purposes including the agricultural sector but not specifically designed to assess agricultural systems (Ciroth, 2007), and they do not readily consider effects of management on soil emissions. Thus, recently, in the area of methodological development, Goglio et al. (2017) adopted five different methods to quantify greenhouse gas emissions of cropping systems in LCA and developed a new tool to perform screening LCAs of cropping systems. On the other hand, in the application of LCA, Yao et al. (2017b) used this method to estimate GHG emission of green manure-based wheat production in China and concluded that introducing legume green manure is a highly efficient measure for persistent GHG reduction, which decreased GHG by 38% for now and 58% in the future. According to LCA method, what an exciting finding we conclude is that crop-livestock integrated system can have a relatively low GHG emissions compared with separated system (no organic wastes returning) (Li et al., 2017), which means that this kind of usage of organic wastes is the better solution to fully run out of the current carbon. Moreover, organic wastes retained in the soil can also increase soil carbon sequestration and the higher C sequestration potential neutralizes the negative impact of organic wastes application on N_2O emissions and results an overall mitigation GWP (Nayak et al., 2015).

3.2. Soil temperature and soil water content affecting soil GHG emissions

About the soil temperature and soil water content, they were not markedly different between treatments, but they were all

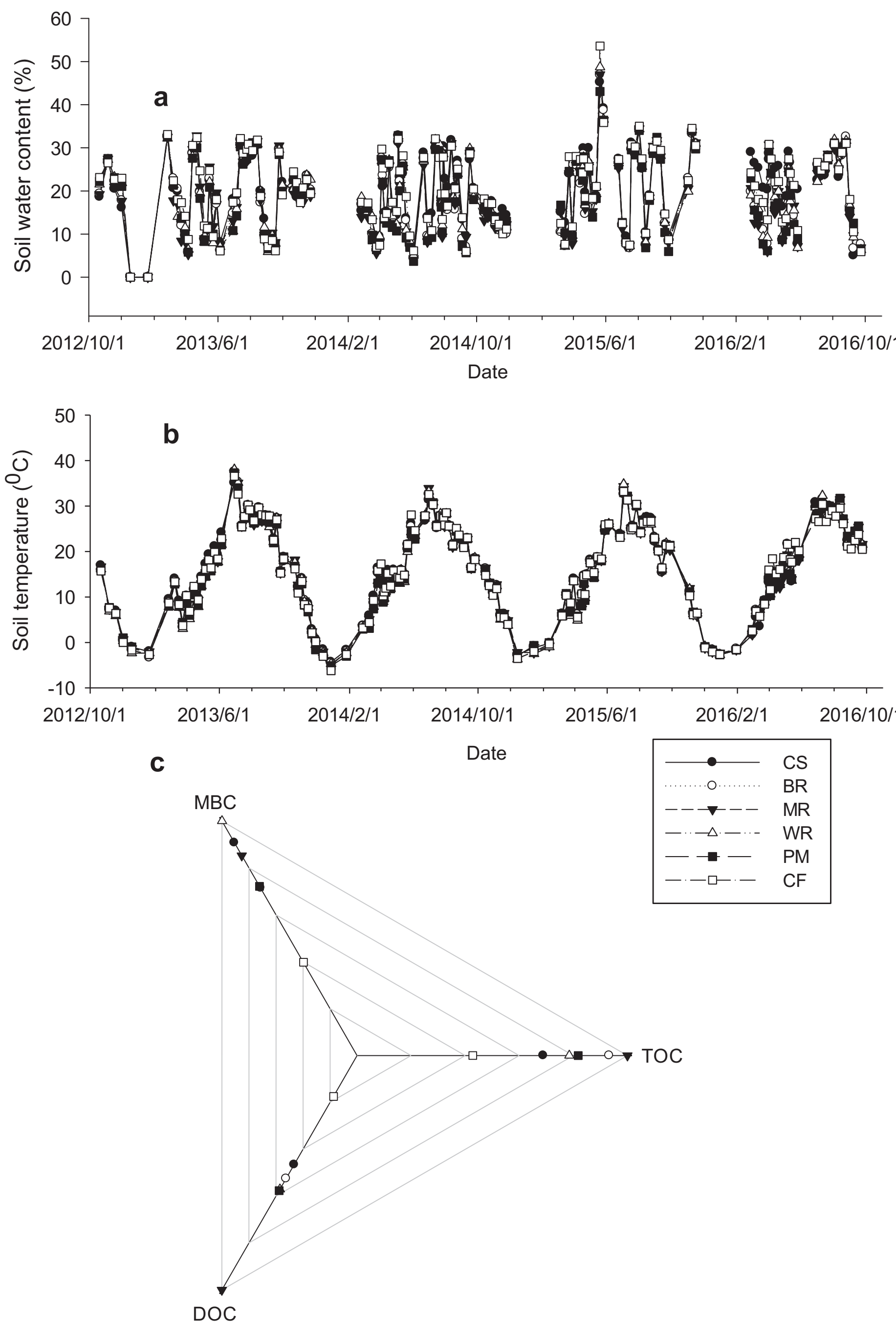

Fig. 2. Soil water content (a), soil temperature (b) and soil TOC, DOC and MBC content(c).

significantly related to soil CO_2 and N_2O. The soil temperature of different material treatments had little differences and the tendencies of them were consistent, partly due to the similar solar radiation, plant density, growth, and canopy shade. The average temperature of CS was the highest, which is 16.81 °C, compared with other organic materials treatments, meaning that crop straw returning might has certain effect of heat preservation (Fig. 2b). Similar with soil temperature, soil water content had no significant difference between different treatments. But there were still some annual characteristics. Firstly, because of the concentrated rainy season from June to September, soil water content in maize season are much higher than wheat season. Secondly, temperature also affected soil water content, more specifically, soil water increased in late February until soil ice melting during the wheat growing season. Secondly, thinking about differences between treatments, soil water content was much lower in the organic waste treatments than CF, which might be due to the plant growing activities, since we indeed had observed that crops growing in organic waste treatments grew better than CF and we know the fact that crops actively growing consume soil water faster (Fig. 2b).

Our results show that soil temperature and soil water content were significantly positively related with soil CO_2 emission in all of the organic wastes treatments as well as CF treatments ($p < 0.01$) and the Pearson correlation coefficients of soil temperature with soil CO_2 emission were higher than that of soil water content, which means that the change of soil temperature will have more powerful potential for soil CO_2 emitting (Table 3). Ding et al. (2007) found that soil temperature is related to soil CO_2 emission, but there is no significant relationship between CO_2 flux and soil moisture. Soil N_2O emissions were significantly positively related to soil water content in all organic wastes treatments besides CF. And the Pearson correlation coefficients of soil N_2O emission with soil temperature were much closer than N_2O emission with soil water content. Paddy field is the most source of CH_4 emission because the covering water provides the ideal condition for anaerobic microorganisms (Smith et al., 2008). However, in the dryland, there are no significant relationship between soil CH_4 emission and soil temperature and soil water content (Table 3).

3.3. Relationships between GHG emissions and soil carbon components

Concentrated on soil carbon components, it is revealed that long term soil amendments with organic wastes increase soil carbon contents and carbon sequestration in soil. Soil TOC content was in the order of BR (8.07 g/kg) > MR (7.86 g/kg) > PM (7.62 g/kg) > WR (6.53 g/kg) > CS (5.81 g/kg), 33.52%, 85.31%, 80.63%, 49.93% and 75.13% greater than that of CF treatment, respectively. Compared with CF treatment, organic wastes also significantly improved soil MBC content ($p < 0.05$). Average results revealed that MBC contents ranged from 100.93 mg/kg to 239.61 mg/kg of each treatment and MBC content of CS, BR, MR and PM were 67.56%, 50.51%, 67.17%, 137.40% and 87.35% greater than that of CF treatment. Soil DOC content of organic wastes treatments were significantly higher than that of CF treatment ($p < 0.05$). Average DOC content of organic wastes treatments is 1.9–3.4 times of CF treatment in wheat season and 2.1–4.4 times of CF treatment in maize season (Fig. 2c).

DOC has already been used as an indicator of C available to soil microorganism, but the relationship between soil CO_2 emission

Table 3
Pearson Correlation coefficient (R^2) between the soil properties and GHG fluxes.

		Factors				
	Treatments	Temperature	Water	TOC	MBC	DOC
CS	CO_2	0.746**	0.373**	0.335*	−0.054	−0.821**
	CH_4	−0.098	0.066	0.050	0.217	−0.628*
	N_2O	0.201*	0.187*	0.274	−0.348	0.745**
BR	CO_2	0.697**	0.402**	0.409*	−0.213	−0.911**
	CH_4	−0.097	−0.027	0.066	−0.121	−0.295
	N_2O	0.192*	0.225**	0.122	−0.335	0.743**
MR	CO_2	0.726**	0.419**	0.427**	0.008	−0.909**
	CH_4	0.045	0.157*	−0.096	−0.03	−0.256
	N_2O	0.191*	0.178*	0.072	−0.196	0.383
WR	CO_2	0.738**	0.442**	0.317	−0.189	−0.874**
	CH_4	−0.156	−0.131	−0.045	−0.146	−0.800**
	N_2O	0.341**	0.205*	0.158	−0.041	−0.39
PM	CO_2	0.680**	0.408**	0.433**	0.355	−0.762**
	CH_4	0.001	−0.100	−0.290	−0.402	0.838**
	N_2O	0.240**	0.237**	0.383*	0.291	0.279
CF	CO_2	0.692**	0.328**	0.661**	0.155	−0.736**
	CH_4	−0.221**	−0.089	−0.317	−0.161	−0.228
	N_2O	0.147	0.131	0.081	−0.119	0.580*

Note: * and ** Significant at $p < 0.05$, and $p < 0.01$, respectively.

Table 4
Characteristic parameters of the organic wastes in this experiment.

Organic materials	C (%)	N (%)	C-N ratio	Cellulose (%)	Hemicellulose (%)	Lignin (%)	Lignin/N	Soluble substance (%)	Ash (%)	Polyphenol (%)
Maize Straw (CS)	43.00	0.90	47.78	31.50	28.61	4.14	4.60	31.00	64.86	0.86
Wheat Straw (CS)	37.87	0.49	77.13	31.76	20.00	6.00	12.22	40.00	54.00	1.06
Biogas Residue (BR)	27.42	1.73	15.90	13.90	14.04	10.73	6.22	50.00	39.27	0.70
Mushroom Residue (MR)	32.33	2.19	14.74	12.74	8.47	22.19	10.11	52.00	25.81	0.71
Wine Residue (WR)	46.10	5.65	8.16	23.25	20.99	9.11	1.61	38.00	52.89	0.65
Pig Manure (PM)	22.43	1.66	13.53	16.22	21.11	5.58	3.37	53.00	41.42	0.69

and DOC is not well understood. Cook and Allan (1992) confirmed that soil respiration rates declined while DOC content remained constant or increased. However, Qiu et al. (2015) reported that application of dissolved organic matter (DOM) stimulated microbial activity, accelerated the decomposition of the applied DOM and increased CO_2 emission. In this study, soil CO_2 emissions were significantly negatively related with soil DOC and the Pearson correlation coefficients were in the range of −0.743 to −0.911, lower than CF treatment, which is −0.736. Nonetheless, CO_2 emissions of organic wastes treatments were significantly positively related to TOC content ($p < 0.05$) except for WR treatment and the Pearson correlation coefficients were in the range of 0.317–0.433, lower than CF treatment, which is 0.661 (Table 3). And the soil CO_2 emissions were significant related with soil MBC. In this study, the results suggested that soil N_2O emissions were only significantly related to DOC just in CS and BR treatments. And since CH_4 is usually formed in soils by microbial breakdown of organic compounds under strictly anaerobic conditions (Smith et al., 2008), CH_4 emission may be correlated to factors but there were no significances between CH_4 emission with these soil carbon components factors (Table 3).

3.4. The effects of composition of organic waste on GHG emissions

Exchange of GHG between soils and the atmosphere was closely linked to organic waste types. There are some obvious relationships between composition properties of organic wastes with GHG emissions, albeit statistical analysis was not conducted because of the shortage of data. Except crop residues, the relatively lower C-N ratio, lignin and polyphenol resulted in higher GHG emissions, while organic wastes with the relatively higher cellulose and hemicellulose caused lower GHG emissions. WR and PM with higher GHG emissions tended to have more cellulose and hemicellulose and less C-N ratio, lignin and polyphenol (Table 4). CS treatment had lower GHG emissions with the fact that organic waste with a C/N ratio in excess of 40 have insufficient N to meet microbial demands for rapid decomposition (Vigil and Kissel, 1991). The results in this paper were consistent with Zou et al. (2005) and Thangarajan et al. (2013) reports, who maintained that emissions are negatively correlated with C/N ratio and lignin content of the incorporated residues. Organic wastes can lead to GHG emission by processes such as priming effect, methanogenesis, nitrification, and denitrification (Thangarajan et al., 2013). And the application of different type of organic wastes affected the distribution of C and N in soil differently, which influences microbial activity and the decomposition rate of plant residues and then affect soil GHG emissions (Qiu et al., 2015). Hence, compared with crop residues, other organic wastes that increased CO_2 emissions might be attributed to the increase of root biomass and microbial activity.

4. Conclusion

Application of organic wastes affected soil GHG emissions significantly. Compared with chemical fertilizer treatments, organic agricultural wastes applied to soil increased GHG emissions. In addition, soil GHG emissions were significantly related to soil temperature, soil water content and DOC in all of the treatments. The analysis of characteristics of organic wastes indicated that there were some relationships between GHG emissions and composition of organic waste, which included C-N ratio, lignin, polyphenol, cellulose and hemicellulose. In this paper, it is recommended that biogas residue returning to soil caused minimum GHG emissions among five different types of organic wastes. And the conclusions will be helpful for supporting policy and decision-making to lead to a more sustainable agriculture with lower GHG emissions. Nevertheless, it is worth noting that except global warming potential, sustainable agricultural production should quantify wide-ranging environmental impacts such as acidification potential, eutrophication potential, cumulative energy demand, toxicity potential and so on. Different wastes were produced by different ways and thus the environmental impacts are different. Future studies should pay more attention to evaluate all of the environmental impacts.

Acknowledgments

This study is supported by the National Natural Science Foundation of China (NNSFC, 31571595) and the National Key R&D Program of the People's Republic of China (2016YFD0300210).

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